Appendix I
Water Quality Criteria

The EPA (1986) has published guidelines for how their criteria are to be applied: “criteria present scientific data and guidance of the environmental effects of pollutants which can be useful to derive regulatory requirements based on consideration of water quality impacts.” Being criteria, they are not legal standards but are indicative of problems that may occur if they are exceeded. However, many states have adopted most of the EPA criteria as enforceable standards. In most cases, the EPA’s criteria are contained in the individual state’s standards. Appropriate water quality criteria is dependent on use classifications.

The following table list typical state water quality criteria for several toxicants (from *Toxic Pollutant Criteria Applicable to State Waters*, Code of Alabama 335-6-10.07). The public water supply and swimming criteria are not shown below.

	Aquatic Life Criteria		Human Life Criteria
	freshwater acute	freshwater chronic	fish consumption only
Arsenic +3	360 µg/L	190 µg/L	-
Arsenic	-	-	(1)
Cadmium	(2)	(2)	-
Chromium +3	(2)	(2)	(3)
Chromium +6	16	11	(3)
Lead	(2)	(2)	-
Mercury	2.4	0.012	(3)
Zinc	(2)	(2)	5,000 µg/L

footnotes:

(1) dependent on cancer potency and bioconcentration factors.
(2) criteria dependent on water hardness.
(3) dependent on reference doses and bioconcentration factors that are developed by the EPA and used by the states.

The Environmental Protection Agency (in *Quality Criteria for Water 1986*, EPA 440/5-86-001) recommends that the acute aquatic life criteria are for one-hour average concentrations that are not to be exceeded more than once every three years, while chronic criteria are for four-day averages that are also not to be exceeded more than once every three years.

If a large percentage of instantaneous observations exceed a criterion, it is apparent, using basic statistical theory, that the observed values are not unique and that longer duration concentrations (such as the one-hour averages and the four-day averages) would also be highly likely to exceed the criterion. Therefore, the frequent exceedences reported in this report are very likely to exist at least for the durations appropriate for the various criteria.

The EPA (in *Quality Criteria for Water 1986*) uses an acceptable exceedence frequency of once per three years because they feel that three years is the average amount of time that it would take an unstressed ecosystem to recover from a pollution event in which exposure to a metal exceeds the criterion. This assumes that a population of organisms exists in adjacent unaffected areas that can recolonize the affected receiving waters. The EPA (also in *Water Quality Criteria*) recommends that total recoverable forms of the metals be compared to the criteria because acid soluble methods have not been approved.

Water Quality Criteria for the Protection of Fish and Wildlife

The following summaries present water quality criteria to protect fish and wildlife resources. Most of this material is from the EPA's *Water Quality Criteria* (1986).

Ammonia

This discussion on the effects of ammonia on aquatic life is a summary from the U.S. EPA's *Quality Criteria for Water, 1986* (EPA 1986). The criteria were published in the Federal Register (50 F.R. 30784, July 29, 1985). The ammonia criteria are only for the protection of aquatic life, as no criteria have been developed for the protection of human health (consumption of contaminated fish or drinking water). The water quality criteria is for general guidance only and do not constitute formal water quality standards. However, the criteria reflect the scientific knowledge concerning the effects of the pollutants and are recommended EPA acceptable limits for aquatic life.

All concentrations used in the water quality criteria report are expressed as un-ionized ammonia (NH_3) because NH_3, not the ammonium ion (NH_4^+), has been demonstrated to be the principal toxic form of ammonia. The amount of the total ammonia (usually expressed as NH_3, but is really a mixture of ionized and un-ionized ammonia forms) that is un-ionized is a function of pH. At low pH values, most of the ammonia is ionized (the ammonium ion, NH_4^+), while at high pH values, most of the ammonia is un-ionized. Therefore, ammonia at high pH values creates more of a problem than similar total ammonia concentrations at low pH values. The un-ionized ammonia concentrations can be calculated, if the pH values are known.

The data used in deriving the EPA criteria are predominantly from flow-through tests in which ammonia concentrations were measured. Ammonia was reported to be acutely toxic to freshwater organisms at concentrations (uncorrected for pH) ranging from 0.53 to 22.8 mg/L NH_3 for 19 invertebrate species representing 14 families and 16 genera and from 0.083 to 4.60 mg/L NH_3 for 29 fish species from 9 families and 18 genera. Among fish species, reported 96-hour LC5O values ranged from 0.083 to 1.09 mg/L for salmonids and from 0.14 to 4.60 mg/L NH_3 for nonsalmonids. Reported data from chronic tests on ammonia with two freshwater invertebrate species, both daphnids, showed effects at concentrations (uncorrected for pH) ranging from 0.304 to 1.2 mg/L NH_3, and with nine freshwater fish species, from five families and seven genera, ranging from 0.0017 to 0.612 mg/L NH_3.

Concentrations of ammonia acutely toxic to fishes may cause loss of equilibrium, hyper-excitability, increased breathing, cardiac output and oxygen uptake, and, in extreme cases, convulsions, coma, and death. At lower concentrations, ammonia has many effects on fishes, including a reduction in hatching success, reduction in growth rate and morphological development, and pathologic changes in tissues of gills, livers, and kidneys.

Several factors have been shown to modify acute NH_3 toxicity in fresh water. Some factors alter the concentration of un-ionized ammonia in the water by affecting the aqueous ammonia equilibrium, and some factors affect the toxicity of un-ionized ammonia itself, either ameliorating or exacerbating the effects of ammonia. Factors that have been shown to affect ammonia toxicity include dissolved oxygen concentration, temperature, pH, previous acclimation to ammonia, fluctuating or intermittent exposures, carbon dioxide concentration, salinity, and the presence of other toxicants.

The most well-studied of these is pH; the acute toxicity of NH_3 has been shown to increase as pH decreases. However, the percentage of the total ammonia that is un-ionized decreases with decreasing pH. Sufficient data exist from toxicity tests conducted at different pH values to formulate a relationship to describe the pH-dependent acute NH_3 toxicity. The very limited amount of data regarding effects of pH on chronic NH_3 toxicity also indicates increasing NH_3 toxicity with decreasing pH, but the data are insufficient to derive a broadly applicable toxicity/pH relationship. Data on temperature effects on acute NH_3 toxicity are limited and somewhat variable, but indications are that NH_3 toxicity to fish is greater as temperature decreases. There is no information available regarding temperature effects on chronic NH_3 toxicity.

Examination of pH and temperature-corrected acute NH_3 toxicity values among species and genera of freshwater organisms showed that invertebrates are generally more tolerant than fishes, a notable exception being the fingernail clam. There is no clear trend among groups of fish; the several most sensitive tested species and genera include representatives from diverse families (Salmonidae, Cyprinidae, Percidae, and Centrarchidae). Available chronic toxicity data for freshwater organisms also indicate invertebrates (cladocerans, one insect species) to be more tolerant than fishes, again with the exception of the fingernail clam. When corrected for the presumed effects of temperature and pH, there is also no clear trend among groups of fish for chronic toxicity values. The most sensitive species, including representatives from five families (Salmonidae, Cyprinidae, Ictaluridae, Centrarchidae, and Catostomidae), have chronic values ranging by not much more than a factor or two. Available data indicate that differences in sensitivities between warm and coldwater families of aquatic organisms are inadequate to warrant discrimination in the national ammonia criterion between bodies of water with “warm” and “coldwater” fishes; rather, effects of organism sensitivities on the criterion are most appropriately handled by site-specific criteria derivation procedures.

Data for concentrations of NH_3 toxic to freshwater phytoplankton and vascular plants, although limited, indicate that freshwater plant species are appreciably more tolerant to NH_3 than are invertebrates or fishes. The ammonia criterion appropriate for the protection of aquatic animals will therefore in all likelihood be sufficiently protective of plant life.

The procedures described in the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if:

(1) the 1-hour* average concentration of un-ionized ammonia (in mg/L NH_3) does not exceed, more often than once every 3 years on the average, the numerical values summarized in the following table, if Salmonids and other sensitive coldwater species are absent:

One-Hour Averaged Maximum Allowable Concentrations for Total Ammonia (mg/L NH_3), For Concurrent pH and Temperature Conditions

pH	$0^{o}C$	$5^{o}C$	$10^{o}C$	$15^{o}C$	$20^{o}C$	$25^{o}C$	$30^{o}C$
6.50	35	33	31	30	29	29	20
6.75	32	30	28	27	27	26	18.6
7.00	28	26	25	24	23	23	16.4
7.25	23	22	20	19.7	19.2	19.0	13.5
7.50	17.4	16.3	15.5	14.9	14.6	14.5	10.3
7.75	12.2	11.4	10.9	10.5	10.3	10.2	7.3
8.00	8.0	7.5	7.1	6.9	6.8	6.8	4.9
8.25	4.5	4.2	4.1	4.0	3.9	4.0	2.9
8.50	2.6	2.4	2.3	2.3	2.3	2.4	1.81
8.75	1.47	1.40	1.37	1.38	1.42	1.52	1.18
9.00	0.86	0.83	0.83	0.86	0.91	1.01	0.82

(*An averaging period of 1 hour may not be appropriate if excursions of concentrations to greater than 1.5 times the average occur during the hour; in such cases, a shorter averaging period may be needed.)

(2) the 4-day average concentration of un-ionized ammonia (in mg/L NH_3) does not exceed, more often than once every 3 years on the average, the average* numerical values summarized in the following table, if Salmonids and other sensitive coldwater species are absent:

Four-Day Averaged Maximum Allowable Concentrations for Total Ammonia ($mg/L\ NH_3$), for Concurrent pH and Temperature Conditions

pH	$0^{o}C$	$5^{o}C$	$10^{o}C$	$15^{o}C$	$20^{o}C$	$25^{o}C$	$30^{o}C$
6.50	2.5	2.4	2.2	2.2	2.1	1.46	1.03
6.75	2.5	2.4	2.2	2.2	2.1	1.47	1.04
7.00	2.5	2.4	2.2	2.2	2.1	1.47	1.04
7.25	2.5	2.4	2.2	2.2	2.1	1.48	1.05
7.50	2.5	2.4	2.2	2.2	2.1	1.49	1.06
7.75	2.3	2.2	2.1	2.0	1.98	1.39	1.00
8.00	1.53	1.44	1.37	1.33	1.31	0.93	0.67
8.25	0.87	0.82	0.78	0.76	0.76	0.54	0.40
8.50	0.49	0.47	0.45	0.44	0.45	0.33	0.25
8.75	0.28	0.27	0.26	0.27	0.27	0.21	0.16
9.00	0.16	0.16	0.16	0.16	0.17	0.14	0.11

(*Because these criteria are nonlinear in pH and temperature, the criterion should be the average of separate evaluations of the formulas reflective of the fluctuations of flow, pH, and temperature within the averaging period; it is not appropriate in general to simply apply the formula to average pH, temperature, and flow.)

The extremes for temperature (0 and $30^{o}C$) and pH (6.5 and 9) given in the above summary tables are absolute. It is not permissible with current data to conduct any extrapolations beyond these limits. In particular, there is reason to believe that appropriate criteria at pH > 9 will be lower than the plateau between pH 8 and 9 shown above. Total ammonia concentrations equivalent to critical un-ionized ammonia concentrations are shown in these tables for receiving waters where salmonids and other sensitive coldwater species are absent. Reported EPA ammonia criteria values for salmonids and coldwater species are the same for temperatures up to $15^{o}C$. For warmer conditions, the total ammonia criteria are about 25% less.

The recommended exceedence frequency of 3 years is the EPA's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to ammonia exceeds the criterion. A stressed system, for example, one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

Nitrates

This discussion on the effects of nitrates on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). These water quality criteria guidance documents do not constitute a national standard.

Two gases (molecular nitrogen and nitrous oxide) and five forms of nongaseous, combined nitrogen (amino and amide groups, ammonium, nitrite, and nitrate) are important in the nitrogen cycle. The amino and amide groups are found in soil organic matter and as constituents of plant and animal protein. The ammonium ion either is released from proteinaceous organic matter and urea, or is synthesized in industrial processes involving atmospheric nitrogen fixation. The nitrite ion is formed from the nitrate or the ammonium ions by certain microorganisms found in soil, water, sewage, and the digestive tract. The nitrate ion is formed by the complete oxidation of ammonium ions by soil or water microorganisms; nitrite is an intermediate product of this nitrification process. In oxygenated natural water systems, nitrite is rapidly oxidized to nitrate. Growing plants assimilate nitrate or ammonium ions and convert them to protein. A process known as denitrification takes place when nitrate containing soils become anaerobic and the conversion to nitrite, molecular nitrogen, or nitrous oxide occurs. Ammonium ions may also be produced in some circumstances.

Among the major point sources of nitrogen entering water bodies are municipal and industrial wastewaters, septic tanks, and feed lot discharges. Nonpoint sources of nitrogen include farm-site fertilizer and animal wastes, lawn fertilizer, sanitary landfill leachatte, atmospheric fallout, nitric oxide and nitrite discharges from automobile exhausts and other combustion processes, and losses from natural sources such as mineralization of soil organic matter (NAS 1972). Water reuse systems in some fish hatcheries employ a nitrification process for ammonia reduction; this may result in exposure of the hatchery fish to elevated levels of nitrite (Russo, *et al*. 1974).

For fingerling rainbow trout, *Salmo gairdneri*, the respective 96-hour and 7-day LC5O toxicity values were 1,360 and 1,060 mg/L nitrate nitrogen in fresh water (Westin 1974). Trama (1954) reported that the 96-hour LC5O for bluegills, *Lepomis macrochirus*, at 20^{o}C was 2,000 mg/L nitrate nitrogen (sodium nitrate) and 420 mg/L nitrate nitrogen (potassium nitrate). Knepp and Arkin (1973) observed that largemouth bass, *Micropterus salmoides* and channel catfish, *Ictalurus punctatus*, could be maintained at concentrations up to 400 mg/L nitrate without significant effect upon their growth and feeding activities.

Nitrite forms of nitrogen were found to be much more toxic than nitrate forms. As an example, the 96-hour and 7-day LC5O values for chinook salmon were found to be 0.9 and 0.7 mg/ L nitrite nitrogen in fresh water (Westin 1974). Smith and Williams (1974) tested the effects of nitrite nitrogen and observed that yearling rainbow trout, *Salmo gairdneri*, suffered a 55 percent mortality after 24 hours at 0.55 mg/L; fingerling rainbow trout suffered a 50 percent mortality after 24 hours of exposure at 1.6 mg/L; and chinook salmon, *Oncorhynchus tshawytscha*, suffered a 40 percent mortality within 24 hours at 0.5 mg/L. There were no mortalities among rainbow trout exposed to 0.15 mg/L nitrite nitrogen for 48 hours. These data indicate that salmonids are more sensitive to nitrite toxicity than are other fish species, e.g., minnows, *Phoxinus laevis*, that suffered a 50 percent mortality within 1.5 hours of exposure to 2,030 mg/L nitrite nitrogen, but required 14 days of exposure for mortality to occur at 10 mg/L (Klingler 1957), and carp, *Cyprinus carpio*, when raised in a water reuse system, tolerated up to 1.8 mg/L nitrite nitrogen (Saeki 1965).

The EPA concluded that (1) levels of nitrate nitrogen at or below 90 mg/L would have no adverse effects on warmwater fish (Knepp and Arkin 1973); (2) nitrite nitrogen at or below 5 mg/L should be protective of most warmwater fish (McCoy 1972); and (3) nitrite nitrogen at or below 0.06 mg/L should be protective of salmonid fishes (Russo, *et al*. 1974; Russo and Thurston 1975). These levels either are not known to occur or would be unlikely to occur in natural surface waters. Recognizing that concentrations of nitrate or nitrite that would exhibit toxic effects on warm- or coldwater fish could rarely occur in nature, restrictive criteria are not recommended.

Phosphate

This discussion on the effects of phosphate on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). These water quality criteria guidance documents do not constitute a national standard.

Phosphorus in the elemental form is very toxic (having an EPA marine life criteria of 0.10 μg/L) and is subject to bioaccumulation in much the same way as mercury. Phosphate forms of phosphorus are a major nutrient required for plant nutrition. In excessive concentrations, phosphates can stimulate plant growth. Excessive growths of aquatic plants (eutrophication) often interfere with water uses and are nuisances to man. Generally, phosphates are not the only cause of eutrophication, but there is substantiating evidence that frequently it is the key element of all of the elements required by freshwater plants (generally, it is present in the least amount relative to need). Therefore, an increase in phosphorus allows use of other already present nutrients for plant growth. In addition, of all of the elements required for plant growth in the water environment, phosphorus is the most easily controlled by man.

Phosphates enter waterways from several different sources. The human body excretes about one pound per year of phosphorus compounds. The use of phosphate detergents increases the per capita contribution to about 3.5 pounds per year of phosphorus compounds. Some industries, such as potato processing, have wastewaters high in phosphates. Many non-point sources (crop, forest, idle, and urban lands) contribute varying amounts of phosphorus compounds to watercourses. This drainage may be surface runoff of rainfall, effluent from agricultural tile lines, or return flow from irrigation. Cattle feedlots, birds, tree leaves, and fallout from the atmosphere all are contributing sources.

Evidence indicates that: (1) high phosphorus compound concentrations are associated with accelerated eutrophication of waters, when other growth-promoting factors are present; (2) aquatic plant problems develop in reservoirs and other standing waters at phosphorus values lower than those critical in flowing streams; (3) reservoirs and lakes collect phosphates from influent streams and store a portion of them within consolidated sediments, thus serving as a phosphate sink; and (4) phosphorus concentrations critical to noxious plant growth vary and nuisance growths may result from a particular concentration of phosphate in one geographical area but not in another. The amount or percentage of inflowing nutrients that may be retained by a lake or reservoir is variable and will depend upon: (1) the nutrient loading to the lake or reservoir; (2) the volume of the euphotic zone; (3) the extent of biological activities; (4) the detention time within a lake basin or the time available for biological activities; and (5) the discharge from the lake.

Once nutrients are discharged into an aquatic ecosystem, their removal is tedious and expensive. Phosphates are used by algae and higher aquatic plants and may be stored in excess of use within the plant cells. With decomposition of the plant cell, some phosphorus may be released immediately through bacterial action for recycling within the biotic community, while the remainder may be deposited with sediments. Much of the material that combines with the consolidated sediments within the lake bottom is bound permanently and will not be recycled into the system.

Although a total phosphorus criterion to control nuisance aquatic growths is not presented, the EPA believes that the following rationale to support such a criterion, which currently is evolving, should be considered.

Total phosphate concentrations in excess of 100 μg/L (expressed as total phosphorus) may interfere with coagulation in water treatment plants. When such concentrations exceed 25 μg/L at the time of the spring turnover on a volume-weighted basis in lakes or reservoirs, they may occasionally stimulate excessive or nuisance growths of algae and other aquatic plants. Algal growths cause undesirable tastes and odors to water, interfere with water treatment, become aesthetically unpleasant, and alter the chemistry of the water supply. They contribute to eutrophication.

To prevent the development of biological nuisances and to control accelerated or cultural eutrophication, total phosphates as phosphorus (P) should not exceed 50 μg/L in any stream at the point where it enters any lake or reservoir, nor 25 μg/L within the lake or reservoir. A desired goal for the prevention of plant nuisances in streams or other flowing waters not discharging directly to lakes or impoundments is 100 μg/L total P (Mackenthun 1973). Most relatively uncontaminated lake districts are known to have surface waters that contain from 10 to 30 μg/L total phosphorus as P (Hutchinson 1957).

The majority of the Nation's eutrophication problems are associated with lakes or reservoirs and currently there are more data to support the establishment of a limiting phosphorus level in those waters than in streams or rivers that do not directly impact such water. There are natural conditions, also, that would dictate the consideration of either a more or less stringent phosphorus level. Eutrophication problems may occur in waters where the phosphorus concentration is less than that indicated above and, obviously, such waters would need more stringent nutrient limits. Likewise, there are those waters within the Nation where phosphorus is not now a limiting nutrient and where the need for phosphorus limits is substantially diminished.

It is evident that a portion of that phosphorus that enters a stream or other flowing waterway eventually will reach a receiving lake or estuary either as a component of the fluid mass, as bed load sediments that are carried downstream, or as floating organic materials that may drift just above the stream's bed or float on its water's surface. Superimposed on the loading from the inflowing waterway, a lake or estuary may receive additional phosphorus as fallout from the atmosphere or as a direct introduction from shoreline areas.

Another method to control the inflow of nutrients, particularly phosphates, into a lake is that of prescribing an annual loading to the receiving water. Vollenweider (1973) suggests total phosphorus (P) loadings, in grams per square meter of surface area per year, that will be a critical level for eutrophic conditions within the receiving waterway for a particular water volume. The mean depth of the lake in meters is divided by the hydraulic detention time in years. Vollenweider's data suggest a range of loading values that should result in oligotrophic lake water quality:

Mean Depth/Hydraulic Detention Time (meters/year)	Oligotrophic or Permissible Loading (grams/meter/year)	Eutrophic or Critical Loading (grams/meter/year)
0.5	0.07	0.14
1.0	0.10	0.20
2.5	0.16	0.32
5.0	0.22	0.45
7.5	0.27	0.55
10.0	0.32	0.63
25.0	0.50	1.00
50.0	0.71	1.41
75.0	0.87	1.73
100.0	1.00	2.00

There may be waterways where higher concentrations, or loadings, of total phosphorus do not produce eutrophication, as well as those waterways where lower concentrations or loadings of total phosphorus may be associated with populations of nuisance organisms. Waters now containing less than the specified amounts of phosphorus should not be degraded by the introduction of additional phosphates

It should be recognized that a number of specific exceptions can occur to reduce the threat of phosphorus as a contributor to lake eutrophication:

1. Naturally occurring phenomena may limit the development of plant nuisances.
2. Technological or cost effective limitations may help control introduced pollutants.
3. Waters may be highly laden with natural silts or colors which reduce the penetration of sunlight needed for plant photosynthesis.
4. Some waters physical features of steep banks, great depth, and substantial flows contribute to a history of no plant problems.
5. Waters may be managed primarily for waterfowl or other wildlife.
6. In some waters, nutrients other than phosphorus (such as nitrogen) is limiting to plant growth; the level and nature of such limiting nutrient would not be expected to increase to an extent that would influence eutrophication.
7. In some waters, phosphorus control cannot be sufficiently effective under present technology to make phosphorus the limiting nutrient.

Dissolved Solids, Conductivity, and Chlorides

This discussion on the effects of total dissolved solids, chlorides, and conductivity on aquatic life and human health is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The water quality criteria guidance documents do not constitute a national standard, but do reflect the scientific knowledge concerning the effects of these pollutants on receiving waters.

Total dissolved solids, chlorides, and conductivity observations are typically used to indicate the magnitude of dissolved minerals in the water. The term total dissolved solids (or dissolved solids) is generally associated with freshwater and refers to the inorganic salts, small amounts of organic matter, and dissolved materials in the water (Sawyer 1960). Salinity is an oceanographic term, and although not precisely equivalent to the total dissolved salt content, it is related (Capurro 1970). Chlorides (not chlorine) are directly related to salinity because of the constant relationship between the major salts in sea water. Conductivity is a measure of the electrical conductivity of water and is also generally related to total dissolved solids, chlorides, or salinity. The principal inorganic anions (negatively charged ions) dissolved in fresh water include the carbonates, chlorides, sulfates, and nitrates (principally in groundwaters); the principal cations (positively charged ions) are sodium, potassium, calcium, and magnesium.

All species of fish and other aquatic life must tolerate a range of dissolved solids concentrations in order to survive under natural conditions. Studies in Saskatchewan found that several common freshwater species survived 10,000 mg/L dissolved solids, that whitefish and pikeperch survived 15,000 mg/L, but only the stickleback survived 20,000 mg/L dissolved solids. It was concluded that lakes with dissolved solids in excess of 15,000 mg/L were unsuitable for most freshwater fishes (Rawson and Moore 1944). The 1968 NTAC Report also recommended maintaining osmotic pressure levels of less than that caused by a 15,000 mg/L solution of sodium chloride.

Indirect effects of excess dissolved solids are primarily the elimination of desirable food plants and other habitat-forming plants. Rapid salinity changes cause plasmolysis of tender leaves and stems because of changes in osmotic pressure. The 1968 NTAC Report recommended the following limits in salinity variation from natural to protect wildlife habitats:

Natural Salinity (parts per thousand)	Variation Permitted (parts per thousand)
0 to 3.5 (freshwater)	1
3.5 to 13.5 (brackish water)	2
13.5 to 35 (seawater)	4

Temperature

This discussion on the effects of temperature is a summary from the U.S. EPA's *Quality Criteria for Water*, 1986 (EPA 1986). These criteria have been previously published by the EPA (*Quality Criteria for Water*, July 1976, PB-263943). The water quality criteria guidance documents do not constitute a national standard, but do reflect the scientific knowledge concerning the effects of these pollutants on receiving waters.

Water temperature affects many beneficial uses, including industrial and domestic water supplies and recreation. The effects of temperature on aquatic life are of the most concern, however, and the water quality criteria were developed to protect the most sensitive aquatic organisms from stress associated with elevated temperatures. Since essentially all of the aquatic organisms are cold blooded, the temperature of the water regulates their metabolism and their ability to survive and reproduce. Temperature, therefore, is an important physical parameter which to some extent regulates many of the beneficial uses of water. The Federal Water Pollution Control Administration in 1967 called temperature "a catalyst, a depressant, an activator, a restrictor, a stimulator, a controller, a killer, one of the most important and most influential water quality characteristics to life in water."

The suitability of water for total body immersion is greatly affected by temperature. In temperate climates, dangers from exposure to low temperatures is more prevalent than exposure to elevated water temperatures. Depending on the amount of activity by the swimmer, comfortable temperatures range from 20^{o} C to 30^{o} C. Short durations of lower and higher temperatures can be tolerated by most individuals. For example, for a 30-minute period, temperatures of 10^{o} C or 35^{o} C can be tolerated without harm by most individuals (NAS 1974).

Temperature also affects the self-purification phenomenon in water bodies and therefore the aesthetic and sanitary qualities that exist. Increased temperatures accelerate the biodegradation of organic material both in the overlying water and in bottom deposits which makes increased demands on the dissolved oxygen resources of a given system. The typical situation is exacerbated by the fact that oxygen becomes less soluble as water temperature increases. Thus, greater demands are exerted on an increasingly scarce resource which may lead to total oxygen depletion and obnoxious septic conditions.

Temperature changes in water bodies can alter the existing aquatic community. The dominance of various phytoplankton groups in specific temperature ranges has been shown. For example, from 20^{o} C to 25^{o} C, diatoms predominated; green algae predominated from 30^{o} C; to 35^{o} C and blue-greens predominated above 35^{o} C (Cairns

1956). Likewise, changes from a coldwater fishery to a warm-water fishery can occur because temperature may be directly lethal to adults or fry, or cause a reduction of activity, or limit their reproduction (Brett 1969).

Upper and lower limits for temperature have been established for many aquatic organisms. Considerably more data exist for upper, as opposed to lower limits. Tabulations of lethal temperatures for fish and other organisms are available (Jones 1964; FWPCA 1967; NAS 1974). Factors such as diet, activity, age, general health, osmotic stress, and even weather contribute to the lethality of temperature. The aquatic species and exposure time are considered the critical factors (Parker and Krenkel 1969).

The effects of sublethal temperatures on metabolism, respiration, behavior, distribution and migration, feeding rate, growth, and reproduction have been summarized by De Sylva (1969). Another study has illustrated that inside the tolerance zone, there is a more restrictive temperature range in which normal activity and growth occur and yet an even more restrictive zone in which normal reproduction will be occur (Brett 1960).

De Sylva (1969) has summarized available data on the combined effects of increased temperature and toxic materials on fish. These data indicate that toxicity generally increases with increased temperature and that organisms subjected to stress from toxic materials are less tolerant of temperature extremes.

The tolerance of organisms to extremes of temperature is a function of their genetic ability to adapt to thermal changes within their characteristic temperature range, the acclimation temperature prior to exposure, and the time of exposure to the elevated temperature (Coutant 1972). True acclimation to changing temperatures requires several days (Brett 1941). Organisms that are acclimated to relatively warm water, when subjected to reduced temperatures that under other conditions of acclimation would not be detrimental, may suffer significant mortality caused by thermal shock (Coutant 1972).

Through the natural changes in climatic conditions, the temperatures of water bodies fluctuate daily, as well as seasonally. These changes do not eliminate indigenous aquatic populations, but affect the existing community structure and the geographic distribution of species. Such temperature changes are necessary to induce the reproductive cycles of aquatic organisms and to regulate other life factors (Mount 1969).

In open waters elevated temperatures may affect periphyton, benthic invertebrates, and fish, in addition to causing shifts in algal dominance. Trembley (1960) studies of the Delaware River downstream from a power plant concluded that the periphyton population was considerably altered by the discharge.

The number and distribution of bottom organisms decrease as water temperatures increase. The upper tolerance limit for a balanced benthic population structure is approximately 32^{o} C. A large number of these invertebrate species are able to tolerate higher temperatures than those required for reproduction (FWPCA 1967).

In order to define criteria for fresh waters, Coutant (1972) cited the following as definable requirements:

1. Maximum sustained temperatures that are consistent with maintaining desirable levels of productivity.
2. Maximum levels of metabolic acclimation to warm temperatures that will permit return to ambient winter temperatures should artificial sources of heat cease.
3. Time-dependent temperature limitations for survival of brief exposures to temperature extremes, both upper and lower.
4. Restricted temperature ranges for various states of reproduction, including (for fish) gametogenesis, spawning migration, release of gametes, development of the embryo, commencement of independent feeding (and other activities) by juveniles, and temperatures required for metamorphosis, emergence, or other activities of lower forms.
5. Thermal limits for diverse species compositions of aquatic communities, particularly where reduction in diversity creates nuisance growths of certain organisms, or where important food sources (food chains) are altered,
6. Thermal requirements of downstream aquatic life (in rivers) where upstream flow reductions of a coldwater resource will adversely affect downstream temperature requirements.

To provide a safety factor, so that none, or only a few, organisms will perish, it has been found experimentally that a criterion of 2^{o} C below maximum temperature is usually sufficient (Black 1953). To provide safety for all the organisms, the temperature causing a median mortality for 50 percent of the population should be calculated and reduced by 2^{o} C in the case of an elevated temperature.

Maximum temperatures for an extensive exposure (e.g., more than 1 week) must be divided into those for warmer periods and winter. Other than for reproduction, the most temperature sensitive life function appears to be growth (Coutant 1972). Coutant (1972) has suggested that a satisfactory estimate of a limiting maximum weekly mean temperature may be an average of the optimum temperature for growth and the temperature for zero net growth.

Because of the difficulty in determining the temperature of zero net growth, essentially the same temperature can be derived by adding to the optimum temperature (for growth or other physiological functions) a factor calculated as onethird of the difference between the ultimate upper lethal temperature and the optimum temperature (NAS 1974).

Since temperature tolerance varies with various states of development of a particular species, the criterion for a particular location should be calculated for the most important life form likely to be present during a particular month. One caveat in using the maximum weekly mean temperature is that the limit for short-term exposure must not be exceeded. Example calculations for predicting the summer maximum temperatures for short-term survival and for extensive exposure for various fish species are presented in Table I-1. These values use data from EPA's Environmental Research Laboratory (ERL) in Duluth.

Table I-1. Maximum Weekly Average Temperatures for Growth, and Short-Term Maxima for Survival for Juveniles and Adults During the Summer (Centigrade and Fahrenheit)

Species	Growth[a]	Maxima[b]
Bluegill	32 (90)	35 (95)
Channel catfish	32 (90)	35 (95)
Largemouth bass	32 (90)	34 (93)

a - Calculated using optimum temperature for growth: maximum weekly average temperature for growth = optimum temperature + 1/3 (ultimate lethal temperature - optimum temperature).

b - Based on acclimation temperature, at the maximum weekly average temperature, needed for summer growth, minus 2^{o} C.

The winter maximum temperature must not exceed the ambient water temperature by more than the amount of change a specimen acclimated to a discharge temperature can tolerate. Such a change could occur by a cessation of the source of heat or by the specimen being driven from an area by high flows, pollutants, or other factors. However, there are inadequate data to estimate a safety factor for the "no stress" level from cold shocks (NAS 1974).

Coutant (1972) has reviewed the effects of temperature on aquatic life reproduction and development. Reproductive events are noted as perhaps the most thermally restricted of all life phases assuming other factors are at or near optimum levels. Natural short-term temperature fluctuations appear to cause reduced reproduction of fish and invertebrates.

There are inadequate data available quantifying the most temperature sensitive life stages among various aquatic species. Uniform elevation of temperature a few degrees, but still within the spawning range, may lead to advanced spawning for spring spawning species and delays for fall spawners. Such changes may not be detrimental, unless asynchrony occurs between newly hatched juveniles and their normal food source. Such asynchrony may be most pronounced among anadromous species, or other migrants, who pass from the warmed area to a normally chilled, unproductive area. Reported temperature data on maximum temperatures for spawning and embryo survival have been summarized in Table I-2 (from ERL-Duluth 1976).

Table I-2. Maximum Weekly Average Temperatures for Spawning and Short-Term Maxima for Embryo Survival During Spawning Season (Centigrade and Fahrenheit)

Species	Spawning[a]	Survival[b]
Bluegill	25 (77)	34 (93)
Channel catfish	27 (81)	29 (84)
Largemouth bass	21 (70)	27 (81)
Threadfin shad	18 (64)	34 (93)

a - The optimum, or mean of the range, of spawning temperatures reported for the species (ERL-Duluth 1976).
b - The upper temperature for successful incubation and hatching reported for the species (ERL-Duluth 1976).

The recommended EPA criteria is in two main parts. The second part is also broken down into four subparts. This detail is needed to account for the differences in temperature tolerance for various aquatic organisms. The EPA criteria are as follows:

For any time of year, there are two upper limiting temperatures for a location (based on the important sensitive species found there at that time):

1. One limit consists of a maximum temperature for short exposures that is time dependent and is given by the species specific equation (example calculated values are shown on Table I-1 under the “maxima” column):

Temperature $=(1/b)[\log(\text{time}) - a] - 2^{o}$ C

where: Temperature is ^{o} C,
exposure time is in minutes,
a= intercept on the “y” or logarithmic axis of the line fitted to experimental data and which is available for some species from Appendix Il-C, National Academy of Sciences 1974 document.
b= slope of the line fitted to experimental data and available for some species from Appendix Il-C, of the National Academy of Sciences 1974 document.

2. The second value is a limit on the weekly average temperature that:

a. In the cooler months (mid-October to mid-April in the north and December to February in the south) will protect against mortality of important species if the elevated plume temperature is suddenly dropped to the ambient temperature, with the limit being the acclimation temperature minus 2^{o} C when the lower lethal threshold temperature equals the ambient water temperature (in some regions this limitation may also be applicable in summer). or

b. In the warmer months (April through October in the north and March through November in the south) is determined by adding to the physiological optimum temperature (usually for growth) a factor calculated as one-third of the difference between the ultimate upper lethal temperature and the optimum temperature for the most sensitive important species (and appropriate life state) that normally is found at that location and time. (Some of these values are given in Table I-1 under the “growth” column). or

c. During reproductive seasons (generally April through June and September through October in the north and March through May and October through November in the south) the limit is that temperature that meets site specific requirements for successful migration, spawning, egg incubation, fry rearing, and other reproductive functions of important species. These local requirements should supersede all other requirements when they are applicable. or

d. There is a site-specific limit that is found necessary to preserve normal species diversity or prevent appearance of nuisance organisms.

Heavy Metals

Many of the heavy metal criteria are defined in terms of water hardness, as elevated water hardness levels have been demonstrated in many laboratory experiments to lessen the toxic effects of these metals. The following tables summarize the applicable criteria, associated with various values of hardness:

Freshwater Aquatic Life Criteria (μg/L)

hardness	Cadmium		Chromium(+3)	
mg/L	acute	chronic	acute	chronic
25	0.82	0.38	560	67
42	1.5	0.57	850	100
54	2.0	0.70	1050	125
63	2.3	0.79	1190	140
74	2.8	0.90	1360	160
84	3.2	0.99	1500	180
90	3.5	1.0	1590	190
98	3.8	1.1	1710	200
110	4.4	1.2	1880	220
120	4.8	1.3	2020	240
140	5.7	1.5	2290	270

Freshwater Aquatic Life Criteria (μg/L) (Cont.)

hardness	Lead		Zinc	
mg/L	acute	chronic	acute	chronic
25	14	0.54	36	33
42	27	1.1	56	51
54	37	1.5	69	63
63	45	1.8	79	72
74	56	2.2	91	82
84	65	2.5	100	91
90	71	2.8	110	97
98	80	3.1	115	100
110	92	3.6	130	115
120	100	4.0	140	120
140	125	4.9	160	140

Hexavalent chromium (Cr^{+6}) and mercury aquatic life problems are not effected by hardness, with the following criteria used to protect aquatic life from exposure to these two metals:

Mercury acute criterion: 2.4 μg/L
Mercury chronic criterion: 0.012 μg/L
Chromium +6 acute criterion: 16 μg/L
Chromium +6 chronic criterion: 11 μg/L

As noted above, the EPA suggests that these aquatic life criteria should not be exceeded more than once every three years. The acute criteria is for a one-hour average, while the chronic criteria is for a four-day average.

Water Quality Criteria for the Protection of Human Health

The following discussion is mostly from the EPA's *Water Quality Criteria* (1986). It summarizes applicable water quality criteria for the protection of human health through both drinking water and fish consumption pathways.

Nitrates
In quantities normally found in food or feed, nitrates become toxic only under conditions in which they are, or may be, reduced to nitrites. Otherwise, at "reasonable" concentrations, nitrates are rapidly excreted in the urine. High intake of nitrates constitutes a hazard primarily to warmblooded animals under conditions that are favorable to reduction to nitrite. Under certain circumstances, nitrate can be reduced to nitrite in the gastrointestinal tract which then reaches the bloodstream and reacts directly with hemoglobin to produce methemoglobin, consequently impairing oxygen transport.

The reaction of nitrite with hemoglobin can be hazardous in infants under three months of age. Serious and occasionally fatal poisonings in infants have occurred following ingestion of untreated well waters shown to contain nitrate at concentrations greater than 10 mg/L nitrate nitrogen (N) (NAS 1974). High nitrate concentrations frequently are found in shallow farm and rural community wells, often as the result of inadequate protection from barnyard drainage or from septic tanks (USPHS 1961; Stewart, *et al*. 1967). Increased concentrations of nitrates also have been found in streams from farm tile drainage in areas of intense fertilization and farm crop production (Harmeson, *et al*. 1971). Approximately 2,000 cases of infant methemoglobinemia have been reported in Europe and North America since 1945; 7 to 8 percent of the affected infants died (Walton 1951; Sattelmacher 1962). Many infants have drunk water in which the nitrate nitrogen content was greater than 10 mg/L without developing methemoglobinemia. Many public water supplies in the United States contain levels that routinely exceed this amount, but only one U.S. case of infant methemoglobinemia associated with a public water supply has ever been reported (Virgil, *et al*. 1965). The differences in susceptibility to methemoglobinemia are not yet understood, but appear to be related to a combination of factors including nitrate concentration, enteric bacteria, and the lower acidity characteristic of the digestive systems of very young mammals. Methemoglobinemia systems and other toxic effects were observed when high nitrate well waters containing pathogenic bacteria were fed to laboratory mammals (Wolff, *et al*. 1972). Conventional water treatment has no significant effect on nitrate removal from water (NAS 1974).

Because of the potential risk of methemoglobinemia to bottlefed infants, and in view of the absence of substantiated physiological effects at nitrate concentrations below 10 mg/L nitrate nitrogen, this level is the criterion for domestic water supplies. Waters with nitrite nitrogen concentrations over 1 mg/L should not be used for infant feeding. Waters with a significant nitrite concentration usually would be heavily polluted and probably bacteriologically unacceptable.

Dissolved Solids, Conductivity, and Chlorides
Excess dissolved solids are objectionable in drinking water because of possible physiological effects, unpalatable mineral tastes, and higher costs because of corrosion or the necessity for additional treatment.

The physiological effects directly related to dissolved solids include laxative effects principally from sodium sulfate and magnesium sulfate and the adverse effect of sodium on certain patients afflicted with cardiac disease and women with toxemia associated with pregnancy. One study was made using data collected from wells in North Dakota. Results from a questionnaire showed that with wells in which sulfates ranged from 1,000 to 1,500 mg/L, 62 percent of the respondents indicated laxative effects associated with consumption of the water. However, nearly one-quarter of the respondents to the questionnaire reported difficulties when concentrations ranged from 200 to 500 mg/L (Moore 1952). To protect transients to an area, a sulfate level of 250 mg/L should afford reasonable protection from laxative effects.

As indicated, sodium frequently is the principal component of dissolved solids. Persons on restricted sodium diets may have an intake restricted from 500 to 1,000 mg/day (National Research Council 1954). The portion ingested in water must be compensated by reduced levels in food ingested so that the total does not exceed the allowable intake. Using certain assumptions of water intake (*e.g*., 2 liters of water consumed per day) and the sodium content of food, it has been calculated that for very restricted sodium diets, 20 mg/L sodium in water would be the maximum, while for moderately restricted diets, 270 mg/L sodium would be the maximum. Specific sodium levels for entire water supplies have not been recommended by the EPA, but various restricted sodium intakes are recommended because: (1) the general population is not adversely affected by sodium, but various restricted sodium intakes are recommended by physicians for a significant portion of the population, and (2) 270 mg/L of sodium is representative of mineralized waters that may be aesthetically unacceptable, but many domestic water supplies exceed this level. Treatment for removal of sodium in water supplies is also costly (NAS 1974).

A study based on consumer surveys in 29 California water systems was made to measure the taste threshold of dissolved salts in water (Bruvold, *et al*. 1969). Systems were selected to eliminate possible interferences from other taste-causing substances besides dissolved salts. The study revealed that consumers rated waters with 320 to 400 mg/L dissolved solids as "excellent" while those with 1,300 mg/L dissolved solids were "unacceptable." A "good" rating was registered for dissolved solids less than 650 to 750 mg/L. The 1962 U.S. Public Health Service Drinking Water Standards recommended a maximum dissolved solids concentration of 500 mg/L, unless more suitable supplies were unavailable.

Specific constituents included in the dissolved solids in water may cause mineral tastes at lower concentrations than other constituents. Chloride ions have frequently been cited as having a low taste threshold in water. Data from Ricter and MacLean (1939) on a taste panel of 53 adults indicated that 61 mg/L NaCl was the median level for detecting a difference from distilled water. At a median concentration of 395 mg/L chloride, a salty taste was identified. Lockhart, *et al*. (1955) when evaluating the effect of chlorides on water used for brewing coffee, found threshold taste concentrations for chloride ranging from 210 mg/L to 310 mg/L, depending on the associated cation. These data indicate that a level of 250 mg/L chlorides is a reasonable maximum level to protect consumers of drinking water.

The EPA criteria for chlorides and sulfates in domestic water supplies is 250 mg/L to protect human welfare.

Heavy Metals

There are also established toxic pollutant criteria for human health protection. These criteria are for carcinogens and non-carcinogens and are established for the consumption of both water and fish and for the consumption of fish only. The equations used by many states to calculate these criteria require that a reference dose and a bioconcentration factor be known for mercury and chromium. A cancer potency factor and a bioconcentration factor is also needed for arsenic, a recognized carcinogen. A risk level of 10^{-5} assumes one increased cancer case per 100,000 people associated with this pollutant and fish consumption. The reference doses and bioconcentration factors are now given by the State of Alabama, for example, in their water quality criteria (Chapter 335-6-10, Appendix A). These values are given by the EPA for 10^{-5}, 10^{-6}, and 10^{-7} risk levels (in *Quality Criteria for Water 1986*). The following list shows these criteria for human health criteria protection for fish consumption only:

Arsenic: 0.175 μg/L (calculated using pg. 39, EPA 1986 values for 10^{-5} risk levels)
Chromium(+3): 3433 mg/L (calculated using pg. 95, EPA 1986 and Alabama values)
Mercury: 0.146 μg/L (calculated using pg. 177, EPA 1986 and Alabama values)
Zinc: 5 mg/L